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Life-cycle studies of biodiesel in Europe: A review addressing the variability of results and modeling issues

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ABSTRACT

Renewable energy sources, and particularly biofuels, are being promoted as possible solutions to address global warming and the depletion of petroleum resources. Nevertheless, significant disagreement and controversies exist regarding the actual benefits of biofuels displacing fossil fuels, as shown by a large number of life-cycle studies that have varying and sometimes contradictory conclusions. This article presents a comprehensive review of life-cycle studies of biodiesel in Europe. Studies have been compared in terms of nonrenewable primary energy requirement and GHG intensity of biodiesel. Recently published studies negate the definite and deterministic advantages for biodiesel presented in former studies. A high variability of results, particularly for biodiesel GHG intensity, with emissions ranging from 15 to 170 $gCO_2eq\ MJ_f^{-1}$ has been observed. A detailed assessment of relevant aspects, including major assumptions, modeling choices and results, has been performed. The main causes for this high variability have been investigated, with emphasis on modeling choices. Key issues found are treatment of co-product and land use modeling, including high uncertainty associated with N_2O and carbon emissions from cultivated soil. Furthermore, a direct correlation between how soil emissions were modeled and increasing values for calculated GHG emission has been found. A robust biodiesel life-cycle modeling has been implemented and the main sources of uncertainty have been investigated to show how uncertainty can be addressed to improve the transparency and reliability of results. Recommendations for further research work concerning the improvement of biofuel life cycle modeling are also presented.

Contents

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1. Introduction

Global warming and depletion of petroleum resources are main concerns in the international agenda. Renewable energy sources, including biofuels, are being promoted as possible contributions to address these problems [1–3]. Nevertheless, significant disagreement and controversies exist regarding the actual benefits of biofuels displacing fossil fuels, as shown by a large number of publications that analyze the life-cycle of biofuels and that have varying and sometimes contradictory conclusions, even for the same biofuel type and pathway [4–10]. This stresses the need to identify the main drivers and to improve the knowledge of the sources for the differences and variations between different studies (and also within specific studies). Are they due to different methodological procedures (or modeling choices), data or production conditions?

The European Union (EU) is the main producer of biodiesel, with close to 65% of the world's production in 2008 [11]. Several issues have been found to affect the calculation of energy and greenhouse gas (GHG) balances of biodiesel, namely: (i) *modeling assumptions* (e.g. approaches used for dealing with biofuel co-products, system boundaries and functional unit, consideration of a reference system for land use) and (ii) *data quality for key input parameters* (e.g. fertilizers and fuel used during raw material cultivation, soil emissions due to land use and land use change). A comprehensive assessment of the key issues that cause variability of the results is needed to ensure reliable outcomes and guarantee the environmental sustainability of the EU policy at this level [8].

This article has three main goals. Firstly, to present a comprehensive review of life-cycle studies published in recent years (since 1998) for biodiesel (Rapeseed Methyl Ester, RME) in Europe. Studies have been compared in terms of nonrenewable primary energy requirement and GHG intensity of biodiesel. A detailed description of relevant aspects, including modeling choices, has been performed to identify the main causes for the high variability of results from the various biodiesel assessments. Secondly, to demonstrate that there is a correlation between the key modeling issues addressed by the surveyed life-cycle models and biodiesel GHG intensity. Thirdly, to show how uncertainty can be addressed in biofuel LC studies, improving the reliability of the results. A life-cycle modeling of biodiesel (RME) in Europe has been performed and the main sources of uncertainty have been investigated: parameter uncertainty and uncertainty concerning how co-product credits are accounted for, namely by expanding the boundaries of the system being studied (expansion and substitution) and using alternative co-product allocation approaches.

The remainder of the article is organized as follows: the next section gives an overview of biodiesel production and consumption in the European Union. Section 3 presents the main findings from a literature review conducted on biodiesel (RME) in Europe from an energy and GHG life-cycle perspective, with special emphasis on the extent to which different modeling choices influence the results. Section 4 presents the LC modeling of biodiesel explicitly addressing uncertainty and discusses the main results in terms of energy requirement and GHG intensity of biodiesel. Section 5 draws the conclusions together and highlights important recommendations.

2. Overview of biodiesel in Europe

2.1. Biodiesel production and consumption

The European Union holds the leading position at worldwide level in terms of biodiesel production [11]. Germany and France are the main biodiesel producers, with a share of nearly 50% of total production in 2009 [12]. The most used raw material is rapeseed, accounting for nearly 84% of the total European biodiesel feedstock [13]. In terms of consumption, biodiesel reached 9.6 million tonnes (t, "metric ton") of oil equivalent (toe) in 2009, which represents 79.5% of the energy content of all biofuels used in European road transport, compared to 19.3% for bioethanol, 0.9% for vegetable oil and 0.4% for biogas [12]. Biodiesel reached a market share of approximately 3.2% in 2009, in terms of total fuel consumption in the European transportation sector, or 4.4% if compared with fossil diesel consumption (assuming a 2.5 ratio of diesel to petrol consumption in Europe [14]). Table 1 gathers information regarding biofuel consumption and market shares in recent years in the EU-27 [12,15,16].

Table 1
Biofuel consumption for transport in the EU-27, including market shares and major biodiesel consumers.

Biodiesel consumption (ktoe)[a]	2005	2006	2007	2008	2009
Germany	1548	2532	2906	2382	2224
France	344	589	1214	1859	2056
United Kingdom	25	132	271	698	823
Italy	172	149	136	658	1049
Spain	23	54	259	520	894
Total biodiesel (EU-27) (ktoe)	2245	4074	5899	8018	9616
Yearly growth (ktoe (%))	–	1829 (81.4%)	1825 (44.8%)	2119 (35.9%)	1598 (19.9%)
Total biofuel (EU-27) (ktoe)	2991	5376	7834	10189	12093
Biodiesel share (%)	75.1	75.8	75.3	78.7	79.5
Biofuels' incorporation rate[b] (%)	1.0	1.8	2.6	3.3	4.0

[a] ktoe: thousand tonnes of oil equivalent.
[b] Biofuel incorporation rate in energy content of total fuel consumption in the transportation sector, as stated in Directive 2003/30/EC [17].

In May 2003, the EU adopted a directive on the promotion of the use of biofuels or other renewable fuels for transport [17]. According to this directive, Member States should ensure that a minimum proportion of biofuels and other renewable fuels is placed on their markets. Specific targets have been set for years 2005 and 2010, respectively 2% and 5.75%, calculated on the basis of energy content of all petrol and diesel marketed for transport purposes. Later, in January 2007, the European Commission proposed "An energy policy for Europe", with the goal to combat climate change and boost the EU's energy security and competitiveness [18]. Based on the European Commission's proposal, in March 2007 the Council endorsed the target of raising the share of biofuels in the transport sector to 10% by 2020. Nevertheless, growing concerns in recent years that the production of biofuels might not respect minimum environmental and social requirements lead to the publication of Directive 2009/28/EC [19] on the promotion of the use of energy from renewable sources. Influenced by the potential negative impacts of biofuels, the EU has broadened the 10% biofuel target: apart from biofuels other renewable energy carriers, such as electricity or hydrogen, may contribute as well to the target. Moreover, compliance with the targets laid down in the directive is only considered for biofuel pathways for which the fulfillment of specific sustainability criteria is demonstrated.

As shown in Table 1, biodiesel consumption in Europe is steadily increasing by 1600–2000 thousand tonnes of oil equivalent per year (ktoe yr^{-1}). Last available data show a 4.0% share by energy content for biofuels in 2009, which indicates that additional efforts are required to reach the target of 5.75% in 2010. The situation among EU member countries is diverse. In Germany, for example, the consumption of biofuels clearly decreased from 2008, after several years of strong growth. This can be explained by the government's decision to reduce tax exemptions and to implement a quota system, as a result of the controversy on the ecological integrity of biofuels and also the costs borne by the German economy [12,15]. In 2009, France has achieved the directive's target of 5.75% and intends to keep raising its incorporation targets for 2010–7.0%—with partial tax exemptions to biodiesel and bioethanol along with authorizations for higher production volumes. Several other countries in the EU-27 are clearly raising the incorporation rates of biofuels in their markets, namely the United Kingdom, Italy and Spain. The growth could be even higher, however, as shown in Fig. 1, in which the evolution of the biodiesel production capacity is compared with actual production data. It can be observed that the gap between capacity and actual production has been growing; according to the European Biodiesel Board [11] approximately 50% of the existing European biodiesel plants remained idle in 2008. This has been caused by market uncertainty together with doubts and controversies concerning biofuels sustainability. For example, the British government has doubts that the directive's objective of 5.75% can be achieved in a

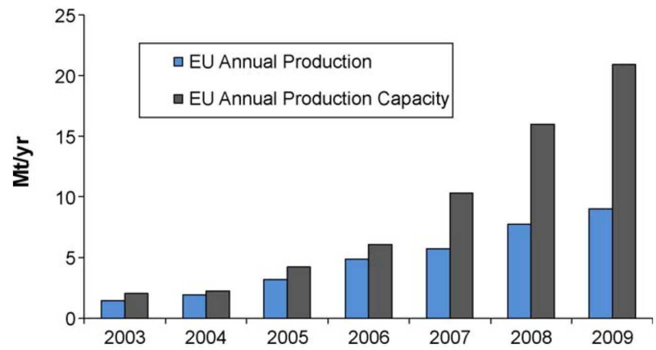


Fig. 1. Biodiesel production capacities and biodiesel production levels in Europe (data from the European Biodiesel Board [11,20]; uncertainty of ±5%).

sustainable and ecological manner and therefore has decided biofuel incorporation volumes in the UK that are below the EU target, as from the 2008/2009 tax year. Nevertheless, the EU installed capacity in 2009 represents an asset to cover an important part of the 10% binding target established by Directive 2009/28/EC. Moreover, it is expected that biodiesel will play an increasingly important role for the EU's security of energy supply, since the petroleum diesel market is subject to a growing deficit [11].

2.2. Biodiesel life-cycle chain

The life-cycle stages of the biodiesel (RME) chain include rapeseed cultivation, harvesting, transport and drying of the seeds, crushing and extraction of the oil, oil degumming and refining, and transesterification. These steps are illustrated in the flowchart of Fig. 2. A detailed description of the rapeseed oil production system (cultivation + oil extraction) can be found, for example, in Malça and Freire [21]. In the transesterification process, the triglyceride molecules of the oil are reacted with methanol in the presence of an alkaline catalyst—to improve the reaction rate and yield—producing a mixture of Rapeseed Methyl Ester and glycerin [22]. After settling, glycerin is left on the bottom and RME is left on top; finally, RME is recovered, washed and filtered [23]. The purpose of transesterication is to lower the viscosity of the oil, improving combustion in diesel engines [24]. Thorough reviews of the use of biodiesel as alternative fuel for diesel engines can be found e.g. in Shahid and Jamal [25] and Murugesan et al. [26].

Two valuable co-products are obtained from the RME production system: rape meal and crude glycerin, as illustrated in Fig. 2. Rape meal is rich in protein and, after a desolventizing process, can be sold for animal feed. Crude glycerin has many potential uses: e.g. replacing grain as animal feed or displacing synthetic glycerin. The biodiesel chain multifunctionality is an important issue which

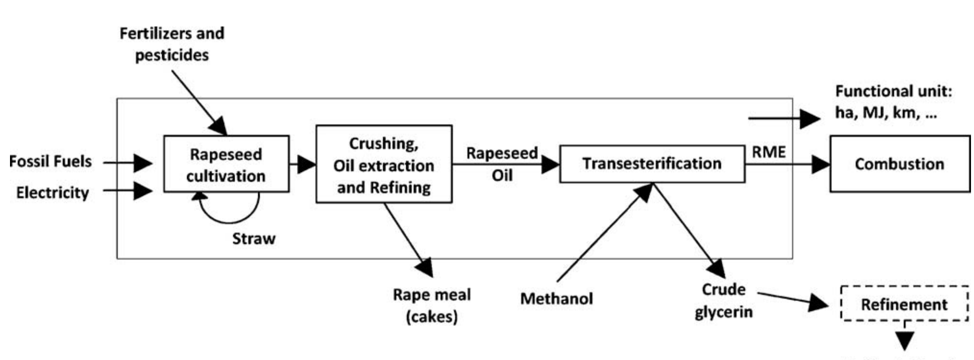


Fig. 2. Flow chart illustrating the life-cycle chain (well-to-tank) of Rapeseed Methyl Ester (RME).

may impact the results of LC studies. Different approaches are available for dealing with biodiesel co-production [27] and this has been a critical issue in biofuel LCA studies, as discussed in Section 3.2.4.

3. Review of biodiesel life-cycle studies for Europe

3.1. Methods

This section presents the main findings from a literature review conducted on life-cycle energy and GHG emissions assessment of rapeseed-based biodiesel (RME) for Europe. An online search of publicly available articles and reports was conducted to find studies that have been published in recent years (since 1998) with detailed information on the methodology, assumptions and data used. A total of more than 40 studies were assessed, of which a selection of 28 is presented in Table 2. The remaining studies have been excluded from Table 2 due to lack of transparency or sufficient quantitative information.

The main results of the surveyed studies in terms of nonrenewable energy requirements (E_{req}) and greenhouse gas (GHG) intensity of biodiesel are summarized in Fig. 3. The nonrenewable primary energy requirement ($E_{req} = \sum E_{nonren,prim}$/FEC) is calculated by evaluating all the nonrenewable energy inputs ($\sum E_{nonren,prim}$) in upstream processing steps like agriculture, transportation and processing, which are compared against the biofuel final energy content (FEC), measured in terms of lower heating value (LHV). The GHG intensity ($gCO_2eq\ MJ_f^{-1}$) quantifies the amount of GHG emissions per unit of FEC. For some surveyed studies, the original outcomes were further calculated to express the results in terms of E_{req} and GHG intensity, as defined above. Studies for which there is a range of results are represented in Fig. 3 by a line connecting the points that define the corresponding lower and upper limits. The correspondence between data labels used in Fig. 3 and surveyed studies is indicated in Table 2. Results from studies that calculated only energy or GHG emissions are shown on the respective axis.

3.2. Comprehensive analysis of surveyed LC studies

3.2.1. Biodiesel nonrenewable energy requirement and GHG emissions

Nonrenewable energy requirement of biodiesel (E_{req}) for the surveyed studies is shown in Fig. 3. Fossil diesel (FD) is also represented and used as a baseline reference. It can be observed that biodiesel E_{req} results present significant variations, ranging from 0.92 $MJ_p\ MJ_f^{-1}$ (the highest value presented in a review performed by the International Energy Agency [28]) to negative energy requirements [29,30]. Negative values can be calculated when energy credits greater than the energy inputs are given to the biodiesel chain: e.g. Bernesson et al. [29] and GM [30], which assume that the co-product glycerin from transesterification displaces the energy intensive production of synthetic glycerin. The E_{req} results for the majority of studies fall in the range of 0.15–0.60 $MJ_p\ MJ_f^{-1}$. This is a broad range, but clearly below the fossil diesel E_{req} meaning that net reductions in fossil energy consumption are obtained when biodiesel displaces fossil diesel. The large variations in biodiesel E_{req} in deterministic studies can be explained by the adoption of different approaches for the treatment of co-products and different assumptions in the agricultural and industrial stages [31–35]. A few studies have included parameter uncertainty, which results in large variations in E_{req} [30,36,37].

Regarding the LC GHG emissions of biodiesel, Fig. 3 shows a very high range of emissions for the surveyed studies, with results from 15 to 170 $gCO_2eq\ MJ_f^{-1}$. This range is broader than the one observed for E_{req} results, particularly when fossil diesel results are taken as a reference. In general, recent studies present higher values—above 60 $gCO_2eq\ MJ_f^{-1}$—which are near or even above fossil diesel GHG intensity. A few recent studies, in particular, show a very high GHG intensity (above 100 $gCO_2eq\ MJ_f^{-1}$) for biodiesel [37,38], which is explained by a very high contribution from carbon and N_2O emissions from soil. Nonetheless, several recent studies also indicate quite low GHG emissions for biodiesel [3,36,39,40].

To facilitate calculation of nonrenewable energy savings and avoided GHG emissions when biodiesel displaces fossil diesel (FD), the latter is also represented in Fig. 3, with 1.136 $MJ_p\ MJ_f^{-1}$ and 82 $gCO_2eq\ MJ_f^{-1}$ (average values presented by Hekkert et al. [41], on the basis of data from 15 studies). Biodiesel studies within the area delimited by dashed lines have both lower GHG intensity and lower E_{req} than fossil diesel.

Results from most former studies report a correlation between biodiesel nonrenewable energy inputs and GHG emissions, as emphasized in the review by Frondel and Peters [42]. However, the results presented in Fig. 3 do not show a general mathematical relationship between GHG intensity and nonrenewable energy requirement. The importance of soil emissions in terms of the overall GHG intensity means that taking into account soil emissions in biofuel LC assessments negates the correlation between nonrenewable energy inputs and GHG emissions presented by most former studies, which did not consider N_2O emissions due to land use and carbon emissions due to land use change (LUC).

The broad range of E_{req} and GHG intensities presented in Fig. 3 stresses the need to understand the main drivers for the differences and variations between different studies (and also within specific studies): are they due to different methodological procedures (or modeling choices), data or production conditions? A comprehensive discussion on the key issues that may affect the life-cycle performance of biofuels follows. These include: geographical scope and system boundaries; functional unit; assessment of co-products; energy and emissions associated with facilities and machinery; reference land use; soil emissions due to land use and land use change; type of LCA approach; and parameter uncertainty [8,38,43,44]. Relevant data from each surveyed study, including major assumptions, methodological choices and results, have been gathered in Table 2. Studies are listed in chronological order.

3.2.2. Geographical scope and system boundaries

The majority of reviewed studies focus on specific European countries, and seven are European-wide assessments. Depending on the study, relevant data for the main stages in biodiesel life-cycle (cultivation of raw materials and industrial conversion) spans from a few years (two or less) to over a decade.

Concerning the system boundaries considered in the studies reviewed in Table 2, different life-cycle approaches have been adopted. The majority of studies (19 out of 28) adopted a "well-to-tank" approach, also called "well-to-pump" [45] or "seed-to-tank" [46]. The "well-to-tank (WtT) assessment considers the steps required to deliver the final (bio)fuel into the on-board tank of a vehicle, namely biomass cultivation, processing, transportation and storage followed by biofuel production, storage and distribution. About 13 studies adopted a full "well-to-wheels" (WtW) assessment. The "well-to-wheels" modeling boundary includes both the "well-to-tank" (WtT) and "tank-to-wheels" (TtW) stages. An example is the JEC [36] detailed report which splits the analysis in the WtT and TtW counterparts and finally aggregates the results in a full WtW assessment. The "tank-to-wheels"—or "pump-to-wheels"—assessment covers only the vehicle operation activities and can be based on data from vehicle simulation models, on-road testing, engine dynamometer experiments or fleet operation data.

Table 2
Surveyed LC studies of biodiesel (RME) production in Europe: relevant data and assumptions, methodological choices and key results.

Surveyed study[a]	De Nocker et al. [76]	IEA [28] [b]	Richards [31]	Scharmer [62]	ADEME[53] [c]	GM [30]	Mortimer et al. [32]	Bernesson et al. [29]	JEC [47]
Data label in Figs. 3 and 4	(98)	(99)	(00)	(01)	(02a)	(02b)	(03)	(04a)	(04b)
Relevant data, choices and assumptions									
Geographical scale	Belgium [d]	n/d	United Kingdom	Europe	France	Europe	UK	Sweden	Europe
Temporal scale	1996–1998	1992–1996	1994–2000 data	1994–2001 data	2002; prospect. up to 2009	1995–1999 data	1996	1990–2001 data	2010–2020
System boundaries	WtW	WtT; WtW	WtT	WtT	WtT[e]	WtT[e,f]	WtT	WtT	WtT; TtW; WtW
Functional unit	kg biodiesel	GJ biodiesel	MJ biodiesel; ha yr	tonne biodiesel	MJ biodiesel	MJ biodiesel	tonne biodiesel	MJ biodiesel	MJ biodiesel; km traveled
Co-product credit approach	No	n/d	Substitution: straw for energy; rape meal as fertilizer; glyc for process energy	Energy allocation + substitution: rape meal as animal feed; glyc for synthetic glyc	Mass allocation + substitution	Substitution: rape meal as animal feed; glycerin as fuel or replacing synthetic glycerin	Economic allocation (rape straw, rape meal and glycerin)	Energy and economic allocation + substitution: rape meal as animal feed; glyc for synthetic glycerin	Substitution: rape meal as animal feed; glycerin for animal feed or propylene glycol
Capital goods	No	No	No	Yes (n/d)	Yes (n/d)	No	1.0% (energy) 0.5% (GHG)	1.4% (energy) 0% (GHG)[l]	No
Agric. ref. system	No	n/d	No	Set-aside	n/d	Set-aside	Set-aside	No	No
Carbon emissions from land use change	No	n/d	No	No	Yes (n/d)	No	No	No	No
N_2O emissions from land use [$kg\,N_2O\,ha^{-1}\,yr^{-1}$]	No	n/d	Yes (1.80)	Yes (3.78)	Yes (0.5% of the N applied)[g]	Yes (4.89; 0.77 min; 13.97 max)	Yes (0.71)	Yes (2.74)	Yes (4.15; 2.91 min; 5.40 max)
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional
Indirect land use change	No	n/d	No	No	No	No	No	No	No
Parameter uncertainty	No	n/d	No	No	No	Yes (Monte-Carlo)	Yes (ranges with upper/lower limits)	Yes (single parameter sensitivity analysis)	Yes (Monte-Carlo)
Selected results									
Energy requirement E_{req} [$MJ_p\,MJ_f^{-1}$]	0.524	0.40; 0.66; 0.92 (min; avg; max)	0.27 (w/ straw); 0.457 (w/o straw)	0.338–0.439[h]	0.334	−0.06 to 0.40 ± 0.01	0.437 ± 0.024 (conv prod) 0.208 ± 0.017 (modified prod)[i]	−0.367; 0.355 (small scale, subst; small scale, economic)	0.44 (glycerin as feed) 0.39 (glyc for glycol)
GHG intensity [$g\,CO_2eq\,MJ_f^{-1}$]	46.7	No	48.2 (w/o straw); 50.5 (w/ straw)	34.4 (w/o soil N_2O); 45.9 (w/ soil N_2O)	20.2–23.7	10.9–77.7	40.6 ± 2.4 (conv prod) 18.8 ± 1.4 (modified prod)	30.9; 51.1 (large scale, subst; small scale, economic)	32.9; 53.9; 73.3 (min; default; max) (glycerin as feed) 28.8; 48.9; 68.9 (glycerin for glycol)

Surveyed study[a]	Janulis [33]	Malça and Freire [57]	Dewulf et al. [64]	SenterNovem [50]	Fredriksson et al. [39]	Lechón et al. [34]	Mortimer and Elsayed [59]	Wagner et al. [49]	JEC [36]
Data label in Figs. 3 and 4	(04c)	(04d)	(05a)	(05b)	(06a)	(06b)	(06c)	(06d)	(07b)
Relevant data, choices and assumptions									
Geographical scale	Lithuania	France	Sweden	Various[j]	Sweden	Spain	UK (North East)	Germany	Europe
Temporal scale	n/d	1993–2004 data	1997/99 data	2005–2008	1994–2004 data	2006	2005	1996–2002 data	2010–2020
System boundaries	WtT	WtT	WtT	WtW	WtW	WtW	WtT	WtT, WtW	WtT; TtW; WtW
Functional Unit	tonne biodiesel	MJ, liter and kg of FD; ha yr	ha yr	km traveled	1000 ha yr	km traveled	tonne biodiesel	kWh; km traveled	MJ biodiesel; km traveled

Co-product credit approach	Energy allocation (straw, rape meal and glycerin)	Mass, energy and economic allocation + Substitution: rape meal as animal feed; glyc for synthetic glycerin	Exergy allocation (straw, rape meal and glycerin)	Economic allocation + substitution: rape meal as animal feed	Economic allocation (co-products n/d)	Economic allocation + substitution: glyc for synthetic glycerin or residue	Economic allocation + substitution: rape meal as animal feed or biomass co-firing[k]	Energy allocation	Substitution: rape meal as animal feed; glycerin for animal feed or propylene glycol
Capital goods	8.8% (energy)	No	No	No	No	No	2.8% (energy) 2.2% (GHG)	No	No
Agric. ref. system	No	Set-aside	No	Set-aside	No	Set-aside	Set-aside	n/d	No
Carbon emissions from land use change	No	No	No	No	No	No	No	n/d	No
N_2O emissions from land use [$kg\,N_2O\,ha^{-1}\,yr^{-1}$]	No	No	No	Yes (2.56–5.60)	Yes (n/d)	Yes (0.4% of the N applied; 0.25% min; 2.25% max)[g]	Yes (4.36)	n/d	Yes (3.12 ± 1.23)
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional
Indirect land use change	No	No	No	No	No	No	No	n/d	No
Parameter uncertainty	No	No	No	Yes (single parameter sensitivity analysis)	Yes (single parameter sensitivity analysis)	Yes (single parameter sensitivity analysis)	Yes (ranges with upper/lower limits)	No	Yes (Monte-Carlo)
Selected results									
Energy requirement E_{req} [$MJ_p\,MJ_f^{-1}$]	0.193–0.446	0.335–0.41	0.324 (exergy basis)	0.49 + 10% 0.49–20%	0.132 + 14.4% 0.132–9.6%[l]	0.184; 0.259; 0.747[m]	0.54 ± 0.026 (animal feed) 0.041 ± 0.03 (biomass co-firing)	0.37	0.51 (glycerin as feed) 0.46 (glyc for glycol)
GHG intensity [$g\,CO_2eq\,MJ_f^{-1}$]	No	13.0–23.0	No	$50.3 \pm 40\%$	22.1 + 26.2% 22.1–13.4%[l]	29.6; 37.6; 60.3	53.8 ± 2.2 (animal feed) 37.5 ± 2.8 (biomass co-firing)	24.9	30.7; 51.8; 68.3 (min; default; max) (glycerin as feed) 25.3; 46.5; 66.6 (glyc for glycol)

Surveyed study[a]	Hansson et al. [40]	Harding et al. [63]	Zah et al. [58]	Halleux et al. [3]	Reijnders and Huijbregts [38]	Stephenson et al. [60]	Lechón et al. [35]	Soimakallio et al. [37]	Thamsiriroj et al. [54]	This study (Section 4)
Data label in Figs. 3 and 4	(07c)	(07d)	(07e)	(08a)	(08b)	(08c)	(09a)	(09b)	(09c)	(10)
Relevant data, choices and assumptions										
Geographical scale	Sweden	n/a	Switzerland[n]	Belgium	Europe	United Kingdom	Spain	Finland	Ireland	Europe
Temporal scale	1994–2004 data	2002/03 data	2004	2005	2002–2007 data	2006/07 data	2008; prospective up to 2020	2003/06 data	2003/06 data	2004–2009
System boundaries	WtW	WtT	WtW	WtW	WtT	WtT	WtW	WtW	WtT[e]	WtT
Functional unit	1000 ha yr	tonne biodiesel	MJ biodiesel; ha yr; person km	100 km traveled	MJ biodiesel; kg biodiesel	tonne biodiesel	km traveled	km traveled	ha yr	MJ biodiesel
Co-product credit approach	Economic allocation (co-products n/d)	Mass allocation (glycerin)	Economic allocation	Substitution: rape meal as animal feed; glycerin for chemicals	Economic allocation (rape meal) (glycerin n/d)	Economic allocation + Substitution: rape meal and glycerin for CHP co-firing	Substitution: rape meal as animal feed; glyc for synthetic glycerin, propylene glycol or residue	Substitution: rape meal as animal feed; glycerin for heat in boilers	No	Mass, energy and economic allocation + substitution: rape meal as animal feed; glycerin as animal feed or replacing synthetic glycerin
Capital goods	No	No	21–27% (GHG)[o]	No	No	4.7–5.9% (energy) 0.9–1.2% (GHG)	No	No	No	No
Agric. ref. system	No	n/a	No	No	No	Set-aside	Set-aside	Set-aside	No	No
Carbon emissions from LUC [$t\,CO_2\,ha^{-1}\,yr^{-1}$]	No	n/d	No	No	Yes (3.08)	No	No	Yes (-0.011–0.286)[p]	No	Yes (0–0.66)
N_2O emissions from land use [$kg\,N_2O\,ha^{-1}\,yr^{-1}$]	Yes (n/d)	n/d	Yes (1.6–3.5% of the N applied)[g]	Yes (n/d)	Yes (2.45–8.20)	Yes (1.60 small scale; 2.11 large scale)	Yes (0.4% of the N applied)[g]	Yes (2.55; 0.40 min; 11.18 max)	Yes (1.70)	Yes (3.12)[q]
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional

Table 2 (*Continued*)

Surveyed study[a]	Hansson et al. [40]	Harding et al. [63]	Zah et al. [58]	Halleux et al. [3]	Reijnders and Huijbregts [38]	Stephenson et al. [60]	Lechón et al. [35]	Soimakallio et al. [37]	Thamsiriroj et al. [54]	This study (Section 4)
Data label in Figs. 3 and 4	(07c)	(07d)	(07e)	(08a)	(08b)	(08c)	(09a)	(09b)	(09c)	(10)
Indirect land use change	No	No	No	No	No	No	No	No	No	No
Parameter uncertainty	No	No	Yes[r]	No	Partially (for N_2O emissions)	No	No	Yes (Monte-Carlo)	No	Yes (Monte-Carlo)
Selected results										
Energy requirement E_{req} [$MJ_p\,MJ_f^{-1}$]	0.120	No	0.68	0.132	0.60[s]	0.538 (small scale) 0.552 (large scale)	0.21 (imported rape) 0.33 (domestic rape)	0.5 ± 0.15	0.456	0.065–0.475 (average) −0.026–0.54 (min; max)
GHG intensity [$g\,CO_2eq\,MJ_f^{-1}$]	21.8	107.3–117.5	50.7; 67.2; 89.5 (min; avg; max)	15.1	123.7–147.8	58.8 (small scale) 64.7 (large scale)	35.4 (imported rape) 76.2 (domestic rape)	80; 100; 170	62.2	21.2–46.4 (average) 2.5–84.6 (min; max)

n/a: not applicable; n/d: not distinguishable.

[a] Each surveyed study is labeled for identification purposes in Figs. 3 and 4.
[b] Several data is not distinguishable because IEA (1999) is a review of different studies.
[c] Only the executive summary was available in the web; therefore, even though a sensitivity analysis has been performed for emissions from cultivated soil, a detailed analysis of the implications of this analysis could not be made.
[d] It also included Western European data, when specific data for Belgium was not available.
[e] Well-to-tank study plus theoretical calculation of combustion on the basis of the carbon content of the fuels.
[f] WtW assessments for different combinations of (bio)fuels and powertrains, but not for rapeseed-based biodiesel, were assumed.
[g] Fertilizer application rates not distinguishable in order to calculate soil N_2O emissions from land use in $kg\,N_2O\,ha^{-1}\,yr^{-1}$.
[h] This study assumes different cultivation locations and different scales for industrial conversion.
[i] Modified, as opposed to conventional, production of biodiesel from oilseed rape consists of low-nitrogen cultivation of oilseed rape, the use of rape straw as an alternative heating fuel in the processing of biodiesel, and the replacement of conventional diesel by biodiesel in agricultural operations and road transport vehicles.
[j] UK, The Netherlands, Germany, France and Poland.
[k] Co-product glycerin was only dealt with by means of economic allocation.
[l] A sensitivity analysis has been conducted to evaluate the effect of ±20% changes in input data, e.g. crop yield, tractor and soil emissions, oil extraction efficiency and oil price.
[m] In addition to different co-product approaches, sensitivity of results has been tested to the origin of rapeseed and the energy efficiency of the industrial conversion stage.
[n] The study covers Swiss and foreign renewable energy production, but only Switzerland for the consumption of renewable energy.
[o] Includes the production and maintenance of vehicles and construction and maintenance of roads.
[p] Negative value means carbon sequestration.
[q] Lognormal distribution (average 3.12 $kg\,N_2O\,ha^{-1}\,yr^{-1}$); standard deviation 3.12 $kg\,N_2O\,ha^{-1}\,yr^{-1}$.
[r] Covers only the uncertainty in the gathering of inventory data.
[s] Cumulative energy demand data (MJ/MJ) from Zah et al. [58] was used.

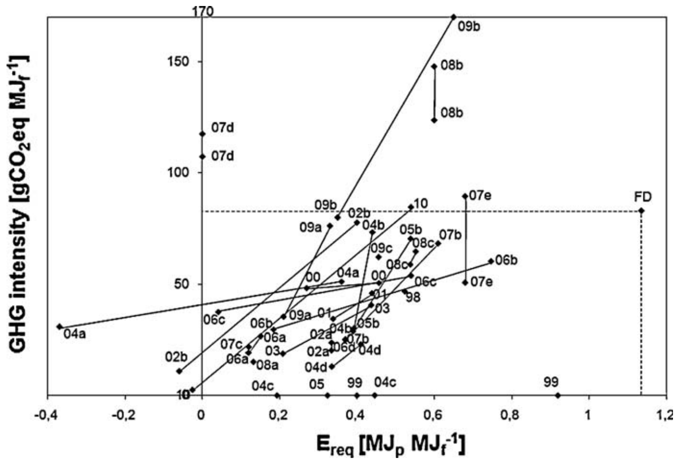


Fig. 3. GHG intensity and nonrenewable energy requirement E_{req} for biodiesel LC studies in Europe (data labels represent surveyed studies; FD: fossil diesel).

The majority of WtW studies assessed the TtW stage using the fuel consumption of a typical passenger vehicle or performing theoretical combustion calculations. In the studies performed by the JEC consortium [36,47], a vehicle simulation tool developed by NREL was used to simulate a compact sized 5-seater European sedan, which enabled the comparison of different (bio)fuels and associated powertrains. Simulation figures were cross-checked with experimental data from a specific top selling model of a European car manufacturer. Lechón et al. [34,35] evaluated the use of different fuels and fuel mixes in a specific vehicle model, which was selected as representative of the Spanish passenger car fleet. The new European driving cycle as defined in [48] was adopted in the study. Tailpipe CO_2 emissions were calculated on the basis of the carbon content of fuels. Other GHG emissions were estimated from literature data and equal emissions were assumed for both biodiesel and fossil diesel combustion. Wagner et al. [49] compared different fuels and propulsion concepts in a medium size automobile operated in the new European driving cycle. The energy efficiency of internal combustion engines running on biofuels and fossil fuels was evaluated on the basis of data from several car manufacturers [49]. SenterNovem [50] combined the passenger car composition for the Netherlands in 2004 with the emission limits for Euro 1-4 specifications [48,51] to estimate average emissions for the Dutch car park. Vehicular emissions from the use of biodiesel and fossil diesel were estimated from van Walwijk et al. [52]. A few other studies in our review also complemented the WtT approach with a theoretical calculation of combustion on the basis of the carbon content of (bio)fuels [30,53,54].

Some authors [7,8] argue that the WtW approach should be the first-choice in LC studies of (bio)fuels, since different fuels may have different engine energy efficiencies. Therefore, the TtW stage should be taken into account and fuels compared for the same transportation service, e.g. distance traveled [7,8]. However, the "well-to-tank" (WtT) assessment is particularly appropriate if the goal and scope is concerned with biodiesel use as a generic energy carrier, without a particular transportation or energy conversion system being considered, which is the case in the majority of reviewed studies. The WtT assessment enables life-cycle inventory results to be analyzed in a variety of different ways, including calculation of potential energy and GHG reductions as well as addressing uncertainty, and avoids the complexities of adding further assumptions, in particular concerning vehicle performance factors, as it is the case when 'kilometers traveled', for example, are adopted as the reference.

3.2.3. Functional units

The definition of a functional unit is an important step in a Life-Cycle Assessment [55]: it is a quantified description of the identified functions (performance characteristics) of a product system and provides a reference to which all other data (inputs and outputs) in the assessment are related [27,55,56]. The definition of the functional unit in biodiesel LC studies is related with the scope and system boundaries of the study; therefore, there is no single or preferred functional unit among reviewed studies. For example, nine studies use 1MJ or 1GJ of fuel energy content (measured in terms of the lower heating value), as this is an appropriate basis for comparison of the energy delivered by a biofuel to the end user. Other studies (7 out of 28) adopt a measure of agricultural surface area (usually the hectare), emphasizing the importance of land use impacts and the scarcity problem of available land for growing energy crops. WtW approaches often use distance traveled (km) as the functional unit [3,34–37,47,49,50].

A few studies use more than one functional unit, which is motivated by different system boundaries or the application of a sensitivity analysis [31,36,47,49,57,58]. As discussed in Section 3.2.2, different system boundaries may be recommended depending on the scope of the study, which may also favor the choice of different functional units.

When required the specific results from each study have been converted to a common functional unit (1 MJ, LHV), based on the specific data included in each reviewed study, so that the outcomes presented in Table 2 are comparable.

3.2.4. Multifunctionality and assessment of co-products

The biodiesel chain is usually multifunctional (i.e. produces more than one product). The studies reviewed used different methods, based on allocation or substitution, to handle multi-functionality. About 18 studies used allocation approaches, on the basis of underlying relationships, to partition the input and output flows of the biodiesel chain between biodiesel and its co-products. The substitution method was used in 16 studies, with various alternative scenarios being adopted. Due to the lack of a common allocation approach among studies a clear trend cannot be identified in the results presented in Fig. 3. Nine studies used both allocation and substitution to handle co-products and three studies did not use any method.

The majority of studies (12 out of 28 studies [29,32,34,38–40,50,57–60]) used economic allocation, in which co-products are allocated according to their market prices. This method is very practical, since it uses the economic value as the main driver [61]. Nevertheless, the volatility of market prices is pointed out as the main drawback of this method, as it may strongly influence the results of the LC study. Other authors prefer relatively fixed physical relationships between co-products, rather than varying economic prices, namely energy [33,49,62], mass [53,63], and exergy [64]. According to ISO 14044 [65], whenever several allocation approaches seem applicable, a sensitivity analysis shall be conducted to illustrate how different methods change the results. However, only Bernesson et al. [29] and Malça and Freire [57] used more than one allocation approach, in order to evaluate the implications of choosing different allocation methods. These authors concluded that the results were largely dependent on the method chosen for allocation of the environmental burdens between biodiesel and its co-products.

Although allocation methods are straightforward to implement, they "arbitrarily" allocate inputs and outputs on the basis of specific relationships between co-products. For this reason, ISO standards on LCA indicate that allocation should be avoided, wherever possible, in favor of subdividing the system in sub-processes (often not possible) or by expanding the system [65], the so-called substitution method (or "avoided-burdens" approach).

Substitution refers to expanding the product system with "avoided" processes (including upstream and downstream links) to remove additional functions related to the functional flows [66].

Sixteen studies in our survey used the substitution method and expanded the biofuel system to include alternative functions for co-products, which are then regarded as credits to the chain. These alternative applications can be diverse, as detailed in Table 2: rape meal is used as fertilizer, animal feed, and in co-firing, whereas glycerin is used for process energy, animal feed, and displacing propylene glycol or synthetic glycerin. Various studies, in particular, assess co-products only through substitution [3,30,31,35–37,47]. According to JEC [36], the substitution approach should be in most cases the preferred method, because it attempts to model reality by tracking the likely fate of co-products. It is therefore important that realistic, as opposed to academic, substitution alternatives are chosen when this method is adopted.

3.2.5. Energy and emissions associated with facilities and machinery

A few studies considered the energy and emissions associated with the construction and maintenance of capital goods. Energy embodied in agromachinery, vehicles and processing plants represents between 1.4% and 8.8% of the total energy requirement for biodiesel production, whereas GHG emissions amount to 0.9–2.2% of the life-cycle GHG intensity of biodiesel [29,33,59,60]. The exception is Zah et al. [58], to which GHG emissions of capital goods represent 21–27% of the LC GHG intensity. This may be explained by the inclusion of a road maintenance stage in the inventory phase. The majority of studies, however, neglected capital goods, acknowledging that they represent only a small fraction of the entire energy and GHG balances.

3.2.6. Reference land use

Several studies (10 out of 28) considered a reference agricultural system consisting of set-aside land to which the rapeseed cultivation system is compared. This hypothesis was in line with European Common Agricultural Policy (CAP) rules in force until 2008, in which set-aside obligations were imposed—farmers were required to leave 10% of their land on set-aside—allowing, however, the cultivation of energy crops on set-aside areas. These obligations, along with a special aid for energy crops of 45€/ha introduced by the 2003 CAP reform, created a favorable environment for the cultivation of energy crops [67]. The set-aside policy changed in 2008, when EU agriculture ministers reached a political agreement on the abolition of compulsory set-aside from 2009 onwards [68], which allowed farmers to maximize their production potential. Primary energy inputs and GHG emissions due to occasional mowing of set-aside areas were taken into account as credits in the biofuel life-cycle studies, since these energy inputs and related emissions would not subsist if the energy crops were cultivated in those areas instead. Three studies [28,49,53] in Table 2 do not indicate if a reference system was taken into account, whereas 13 studies simply did not consider any reference agricultural system, mainly because rapeseed cultivation was assumed to be within a crop rotation scheme.

3.2.7. Carbon emissions due to land use change

Of the twenty-eight reviewed studies, only four considered the contribution of soil carbon emissions for the GHG balance. Soimakallio et al. [37] used IPCC [69] data for calculating the annual change in soil carbon balance during 100 years; upper and lower limits were considered for conventional tillage and no-tillage cultivation of rapeseed, respectively. Vleeshouwers and Verhagen [70] developed a model to calculate carbon fluxes from agricultural soils in Europe, which includes the effects of crop, climate and soil on the carbon budget of agricultural land. According to these authors, European arable soils are estimated to lose 0.84 ± 0.40 t C ha^{-1} yr^{-1} [70,71]. In the work by Reijnders and Huijbregts [38], which is included in our review, the average value of 0.84 t C ha^{-1} yr^{-1} (3.08 t CO_2 ha^{-1} yr^{-1}) was used for rapeseed cultivation in a crop rotation system.

Vleeshouwers and Verhagen [70], Reijnders and Huijbregts [38] and Soimakallio et al. [37] concluded that direct soil carbon emissions from land use change is an important aspect for the GHG balance of biofuels. However, our review shows that this issue has not captured enough attention, even in recent biofuel LC studies. It must be also emphasized that carbon emissions due to land use change are intimately correlated with the reference land use considered, as demonstrated e.g. by Hoefnagels et al. [72] and Malça and Freire [73].

3.2.8. N_2O emissions due to land use

Even though N_2O emissions from soil were taken into account in the majority of reviewed studies (21 out of 28), this assessment was in most cases performed with deterministic data. Several authors estimated nitrous oxide emissions using single figures for the N_2O emitted, which was calculated as a percentage of the N fertilizer input to cultivated soil [29,31,32,35,53,54,59,60,62]. A wide range of deterministic values was used across studies (see details in Table 2).

The assessment of nitrous oxide emissions from soil has recently proven to be an important issue in the GHG balance of biofuels, e.g. [38,74]. These emissions vary widely and depend upon a number of factors such as soil type, climate, tillage, fertilizer rates and crop type [4,38,60,74,75]. A few studies in our review included uncertainty data concerning N_2O emissions, whether through ranges with upper and lower limits [34,50], or by using stochastic methods [30,36,37,47]. The uncertainty ranges used are listed in Table 2.

3.2.9. Type of LCA approach

More recently, two different approaches to LCA have been proposed: attributional (or retrospective) LCA and consequential (or prospective) LCA [27]. All surveyed LC studies in Table 2 are attributional. The attributional approach for LCA aims at describing environmentally relevant physical flows to and from a life-cycle and its sub-systems and therefore uses average data. Prospective or consequential LCA aims at assessing the consequences of change compared to the present situation, that is how the environmentally relevant physical flows to and from the life-cycle will change in response to possible changes; therefore, consequential LCA uses marginal data [77,78]. Nonetheless, consequential LCA is still in its earliest stages of development, and a reliable methodology has yet not been established for bioenergy studies [79].

An aspect that requires a consequential approach in LC studies is the assessment of indirect land use change issues associated with biofuels. Increased biofuels demand may lead to an expansion of cropped area at the expenses of other land uses. The displacement of prior crop production to other areas (indirect LUC) may contribute to important environmental impacts, namely GHG emissions [80–82], which has recently been the subject of important controversy among the scientific community. This builds on the fact that market mechanisms should be taken into account when modeling all the consequences of increased consumption of biofuels, which requires subjective assumptions and leads to potentially higher complexity and uncertainty. Further work is thus required to address the practical modeling of indirect LUC associated to biofuels, as stated e.g. by Anex and Lifset [79] and Kløverpris et al. [83].

3.2.10. Parameter uncertainty

Concerning the inclusion of parameter uncertainty in surveyed studies, it can be seen that former studies did not consider this type

of uncertainty [31,33,40,49,53,57,62–64,76] or, at least, it was only considered in a simplified way through single parameter sensitivity analysis [29,32,34,39,50,59]. The exceptions are GM [30] and JEC [47], in which parameter uncertainty was evaluated using Monte-Carlo simulation, a technique that proves difficult in becoming standard, as recent studies that still do not include parameter uncertainty have show [3,35,54,60]. The adoption of probabilistic approaches to address previously neglected issues, namely soil emissions with high uncertainty [37], leads to higher GHG emissions and wider biodiesel GHG ranges. Moreover, the conclusion of some former studies indicating that the results of biofuel LC studies were largely dependent on the allocation method selected for co-product evaluation can be questioned when parameter uncertainty is included in the assessment, as recently demonstrated by Malça and Freire [10] for the production of vegetable oil from rapeseed in Europe.

3.2.11. Former review studies

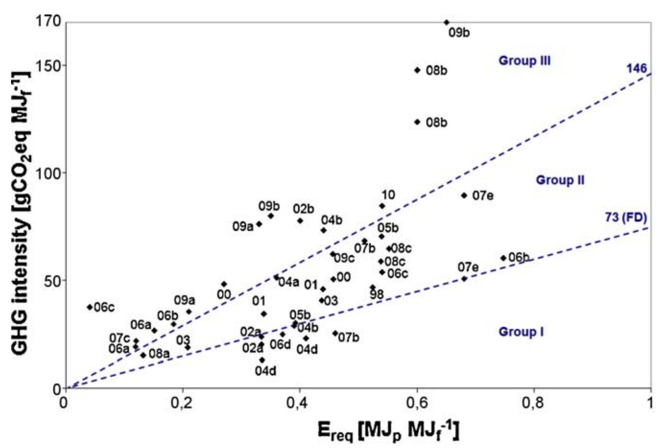
Most former studies presented clear advantages in terms of GHG intensity for biodiesel over fossil diesel because they neglected carbon emissions from soils and were based on deterministic life-cycle models. This is the case with all surveyed studies up to 2006 in our review, with the exception of GM [30] and JEC [47]. Other studies in the literature point out the same conclusion. For example, the International Energy Agency conducted a review of several studies, dated from 1993 to 2002, on the energy requirements and well-to-wheels GHG emission impacts from using rapeseed-derived biodiesel rather than conventional diesel fuel [84]. Main findings from this survey were that fossil energy requirement of biodiesel production systems vary between 0.33 and 0.57 MJ per MJ of biofuel energy content. The estimates for net GHG emission reductions in light-duty compression-ignition engines range from 44% to 66%. Richards [31] also concluded that biodiesel production was strongly energy positive and, where straw was burned as fuel and oil seed rape meal used as a fertilizer, the balance was even better. Larson [4] conducted a review of several LC studies covering a variety of conventional and future generation liquid biofuels for transportation, in which different aspects are highlighted that justify the wide range of results between studies. Due to the broadening scope of the study, only a few studies addressing rapeseed-based biodiesel have been assessed in the review; for these, the main finding was that RME shows GHG emission savings compared to conventional diesel fuel. Frondel and Peters [42] also found that the energy and GHG balances of supporting rapeseed-based biodiesel as a substitute for fossil diesel were clearly positive. Based on a survey of empirical studies, these authors concluded that between 55% and 79% of fossil resources can be saved with the substitution. Moreover, those authors found that GHG balances were intimately correlated with energy balances, with estimates of GHG savings in the range of 41–78%. Recently, Yan and Crookes [85] have published a review of nine studies addressing the life-cycle of rapeseed-derived biodiesel. Depending on the study, these authors concluded that the fossil fuel use and GHG emissions for biodiesel were in the range of, respectively, 0.33–0.65 $MJ_p\,MJ_f^{-1}$ and 20–53 $gCO_2eq\,MJ_f^{-1}$. This low level of emissions may be explained by deterministic assessments not accounting for N_2O or carbon emissions from soil.

Hoefnagels et al. [72] reviewed the impact of different assumptions and methodological choices on the life-cycle GHG emissions of various biofuels (bioethanol, biodiesel and Fischer–Tropsch diesel). Key factors affecting the performance of biofuels included allocation of co-products, location of crop cultivation production, crop yields, reference land (LUC) and soil N_2O emissions. Concerning rapeseed-based biodiesel (RME), only one study was reviewed. The main conclusion is that RME GHG emissions can vary between 17 and 140 $gCO_2eq\,MJ_f^{-1}$ depending on the key parameters and methodological choices considered. Concerning carbon emissions from LUC, Majer et al. [86] conducted a review of biodiesel LC studies and showed the significant influence of LUC effects on the potential GHG emission savings associated with biodiesel from palm and jatropha. None of the rapeseed-based biodiesel studies in the revision by Majer et al. [86] addressed land use change issues, however.

4. Discussion

Fig. 4 groups the surveyed studies according to the extent to which some of the key methodological GHG issues have been addressed in the reviewed assessments, namely inclusion of N_2O and carbon emissions from cultivated soil. All reviewed studies take into account fossil CO_2 emissions throughout the life-cycle, but do not follow the same methodology concerning soil emissions. Fig. 4 shows that a direct linkage exists between taking into account soil emissions in biofuel life-cycle studies and increasing values for calculated GHG emissions. Studies have been divided into three groups: Group I gathers studies that do not account for N_2O emissions from soil or at most adopt low (and deterministic) values for these emissions. Group II includes studies that account for higher N_2O emissions from soil. Group III addresses the studies that include the additional assessment of soil carbon emissions, in addition to higher nitrous oxide emissions with important uncertainty ranges.

As shown in Fig. 4, the classification in three groups can also be made in terms of the GHG intensity per nonrenewable energy use requirement. Dashed lines in Fig. 4 indicate the thresholds considered for grouping: Group I—studies with values below that of fossil diesel (73 $g\,CO_2eq\,MJ_p^{-1}$ [41,87]); Group II—values between 73 and 146 $g\,CO_2eq\,MJ_p^{-1}$; and Group III—values above 146 $g\,CO_2eq\,MJ_p^{-1}$ (twice the value of FD). Our results show that in Group I studies, there is a correlation between biodiesel nonrenewable energy inputs and GHG emissions, close to the fossil diesel value of 73 $g\,CO_2eq\,MJ_p^{-1}$. This means that GHG emissions are mainly due to fossil energy use. Values lower that



Group	Key modeling issues addressed	GHG intensity per nonrenewable primary energy use requirement ($gCO_2eq\,MJ_p^{-1}$)		
		< 73	73 – 146	> 146
III	soil carbon emissions + soil N_2O emissions (↑, high uncert.) + fossil CO_2 emissions			10, 09b, 08b
II	soil N_2O emissions (↓ or ↑) + fossil CO_2 emissions		09c, 08c, 08a, 07e, 07b, 06c, 06b, 05b, 04b, 04a, 03, 01, 00	09a, 07c, 06c, 06a, 02b
I	soil N_2O emissions (0 or ↓) + fossil CO_2 emissions	06d, 04d, 02a	98	

Fig. 4. GHG intensity and nonrenewable energy use requirement; FD: fossil diesel. Data labels are defined in Table 2.

73 g CO_2eq MJ_p^{-1} may indicate that the mix of nonrenewable energy used is less GHG intensive than fossil diesel, has it happens e.g. for biodiesel produced in France (nuclear energy) [57]. It should be noted that Group I mainly includes former studies (up to 2006).

Groups 2 and 3 report in general more recent assessments in which further key methodological issues concerning GHG emissions not related with energy use are addressed, namely N_2O and carbon emissions from cultivated soil. In these studies, the GHG emissions per nonrenewable MJ_p are considerably higher than those for fossil fuels, since GHG emissions are not exclusively linked to energy use. Soil emissions take the lead over energy use in terms of the critical factor for overall GHG emissions. This is particularly notorious in Group 3, for which GHG emissions per nonrenewable MJ_p more than double those from fossil energy use.

Recently published studies negate the definite and deterministic advantages for biodiesel presented in former studies. The reason is twofold: recent studies have included soil emissions (mainly N_2O and, not as often, carbon associated with LUC, Sections 3.2.7 and 3.2.8) and have taken into account uncertainty related to parameters. In our review, Soimakallio et al. [37] and Reijnders and Huijbregts [38] present very high biodiesel GHG emissions, much higher than for the other assessed studies, which is due to high GHG emissions from soil with significant uncertainty. Even though direct carbon emissions from land use change may strongly contribute for the GHG balance of biofuels, our review shows that this issue has not been commonly addressed, even in recent biofuel LC studies.

Another important conclusion from our review negates the correlation between biodiesel nonrenewable energy inputs and GHG emissions reported in most former studies. Results presented in Fig. 3 do not show a general mathematical relationship between GHG intensity and nonrenewable energy requirements. The importance of soil emissions in terms of the overall GHG intensity means that taking into account soil emissions in biofuel LC assessments negates the correlation between nonrenewable energy inputs and GHG emissions presented by most former studies, which did not consider N_2O emissions due to land use and carbon emissions due to LUC. Therefore energy cannot be used as a proxy for emissions as also shown in [37,38,88].

This review shows how different key issues in LC studies of biodiesel affect the outcomes in terms of primary energy consumption and greenhouse gas emissions. It has been demonstrated that taking account of parameter uncertainty for certain key inputs (e.g. N_2O and carbon emissions from soil), as well as selection of different options for dealing with co-products (scenario uncertainty), has a strong influence in the results. In particular, last column of Table 2 lists selected data and results from our own LC study for biodiesel in Europe. Our study is presented in Section 4, with emphasis on discussing how we have addressed uncertainty issues and their importance in the energy and environmental modeling of biodiesel.

5. Biodiesel life-cycle modeling addressing uncertainty

5.1. Goal, scope and main assumptions

This section shows how parameter and scenario uncertainties can be addressed in the calculation of the LC energy requirement and GHG intensity of biodiesel from rapeseed in Europe. Parameter and scenario uncertainty are also compared against each other, in order to evaluate the relative importance. This approach is useful in determining on what sources of uncertainty to improve our knowledge in order to further reduce the overall uncertainty of a LC study.

A "well-to-tank" approach has been used (cf. Fig. 2) to assess energy and GHG emissions of biodiesel. The functional unit chosen is 1 MJ of Fuel Energy Content (FEC), measured in terms of the lower heating value. The greenhouse gases considered are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). It was found that other GHG occur in negligible amounts in the biofuel system analyzed and were, therefore, not followed up. An uncertainty of ±35% for the 90% confidence range has been considered for the global warming potentials (GWP) of CH_4 and N_2O, according to IPCC [89]. N_2O emissions caused by cultivation have been estimated on the basis of data from [59,69,74,90,91].

Concerning direct carbon emissions associated with LUC, different sources have been identified that evaluate the variation in soil carbon content when land is converted from a previous crop to rapeseed cultivation [69,86,92–97]. From the available data, a uniform probability distribution between 0 t C ha^{-1} yr^{-1} (min) and 0.18 t C ha^{-1} yr^{-1} (max) has been considered for soil carbon release in the stochastic GHG emissions calculation addressing uncertainty. It has been assumed that the carbon released from soil oxidizes, which increases the concentration of carbon dioxide in the atmosphere [69,86,88].

5.2. Addressing uncertainty

Monte Carlo simulation has been used to quantify parameter uncertainty; this technique propagates known parameter uncertainties into uncertainty distributions of the output variables [98]. Parameter values and probability distributions for Monte-Carlo uncertainty propagation concerning the various processes represented in the flowchart of Fig. 2 can be found in Malça and Freire [10,99]. A random sampling procedure and 10000 iterations per simulation have been used. The probability distributions of the outcomes have been divided in the 5th, 25th, 50th, 75th, and 95th percentiles and are displayed in box charts.

Different approaches have been addressed concerning the modeling choice of how co-product credits are accounted for [65,66], namely allocation and the substitution method. The mass, energy and economic parameters used for allocation of co-products can be found in Malça and Freire [10,99], on the basis of data from [29,36,59]. Concerning the application of the substitution method, it has been considered that rapeseed meal from oilseed crushing is replacing soybean meal imports as a high-protein animal feed [100,101]. The substitution options considered for glycerin include replacing grain as animal feed and displacing synthetic glycerin, as they are considered, respectively lower and upper limits in terms of energy and emission credits given to the biodiesel chain [36,82].

5.3. Results: energy requirement and GHG intensity

The energy requirement E_{req} of biodiesel is presented in Fig. 5 for various alternative co-product evaluation methods—scenario

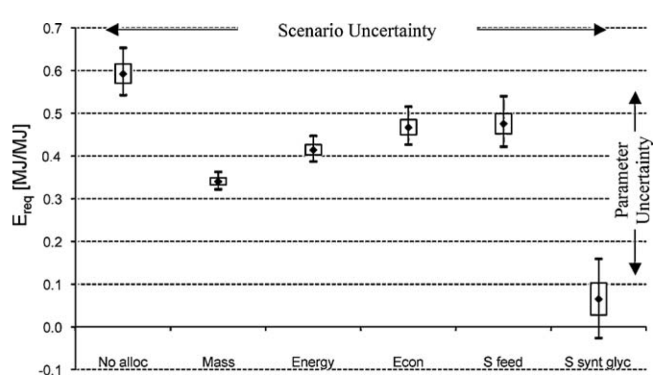


Fig. 5. Biodiesel E_{req} results: scenario and parameter uncertainty. "S" stands for Substitution method. The same notation is used in Fig. 6.

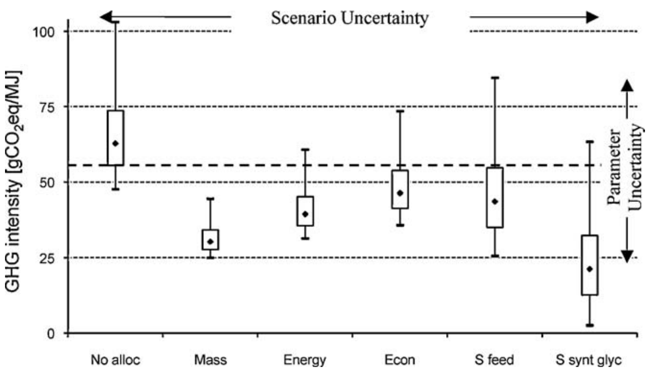


Fig. 6. Biodiesel GHG intensity: scenario and parameter uncertainty (100-year time horizon).

uncertainty. The "no allocation" approach, in which co-products are not taken into account and all energy inputs are allocated to the main product RME, and the synthetic glycerin substitution method which brings very high energy credits, represent, respectively, the lower and upper limits of energy credits given to the RME chain. For the other options, average E_{req} values are likely to remain within the range 0.25–0.50 $MJ_p\,MJ_f^{-1}$. Although parameter uncertainty is higher in the case of substitution due to higher uncertainties associated with the displaced products, Fig. 5 shows that scenario uncertainty is still more significant than parameter uncertainty when assessing biodiesel energy requirement.

The GHG intensity of biodiesel is presented in Fig. 6. The main difference to the energy results of Fig. 5 is the high level of parameter uncertainty associated with GHG emissions, which is mainly due to nitrous oxide and carbon emissions from soil. The ranges of GHG emissions calculated for each scenario clearly overcome the differences between scenarios. In particular, the adoption of the substitution method introduces high uncertainty, through the substitution options selected for rape meal and glycerin. Therefore, it is difficult to definitely point out a most favorable option on the basis of a GHG emissions criterion. Furthermore, when biodiesel GHG emissions are compared with the threshold set in the European directive 2009/28/EC [19] (GHG emission savings of 35% over fossil diesel, dashed line in Fig. 6), a probability exists that biodiesel emissions stay above that limit. This is valid for all scenarios, including the substitution approach and the energy allocation, which are the methods indicated in the directive. The exception is mass-based allocation. It must be emphasized, however, that despite being a straightforward method, mass-based allocation is very often a meaningless approach, namely when energy systems or market principles come into play.

An uncertainty contribution analysis has been performed, showing that fuel used in agricultural machinery and nitrogen fertilizer are the main contributors to the variance of biodiesel E_{req}. In terms of emissions, nitrous oxide released from the use of fertilizers and CO_2 emissions from soil carbon stock changes are the most important parameters affecting the variance of biodiesel GHG intensity.

6. Conclusions and recommendations

A comprehensive review of published life-cycle studies for biodiesel from rapeseed in Europe has been performed. A high variability of results, particularly for biodiesel GHG intensity, with emissions ranging from 15 to 170 $gCO_2eq\,MJ_f^{-1}$ has been observed. The main causes for this high variability have been investigated, with emphasis on modeling choices. Key issues found are treatment of co-product and land use modeling, including high uncertainty associated with N_2O and carbon emissions from cultivated soil. Furthermore, a direct correlation between how soil emissions were modeled and increasing values for calculated GHG emission has been found for the surveyed studies.

Our review also shows a time-dependent evolution of results: more recent assessments show higher GHG intensity and variability than former studies, due to evolving GHG modeling approaches used in biofuel life-cycle studies. Most former studies in our review show clear advantages for biodiesel over fossil diesel in terms of life-cycle GHG intensity. Moreover, they report a correlation between biodiesel nonrenewable energy inputs and GHG emissions. Other studies in the literature point out the same conclusion. However, we demonstrate that taking into account soil emissions in biofuel LC assessments, namely N_2O emissions due to land use and carbon emissions due to land use change, negates that correlation. Soil emissions are not exclusively linked to energy use; hence, energy cannot be used as a proxy for emissions. Our review also shows that soil emissions take the lead over energy use in terms of the critical factor for the overall GHG intensity of biodiesel. In particular, taking account of parameter uncertainty for soil emissions strongly affects the GHG emission results of biodiesel. This conclusion draws from our biodiesel life-cycle modeling addressing uncertainty, in which several sources of uncertainty have been investigated.

Our work highlights the need for transparency in assumptions and inputs to LC models and demonstrates that neglecting key issues—and related uncertainty—in the life-cycle GHG accounting of biodiesel may compromise the reliability of results. As a general recommendation: outputs from LC studies need to be validated and verified. N_2O and carbon emissions from cultivated soil have a substantial effect on biodiesel GHG intensity and require further research efforts to improve. It is important to incorporate uncertainty analysis in the life-cycle modeling of biofuels, in order to reduce the uncertainty level in the results and to better support decisions on whether or not to promote specific biofuel pathways.

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